Working Paper

Water Quality Management of the Nitra River Basin (Slovakia): Evaluation of Various Control Strategies

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WP-93-63
October 1993

IIASA

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Preface

The development of water quality management strategies of river basins involves the identification and evaluation of the appropriate control alternatives in order to satisfy various water quality, economic, and other goals. IIASA's Water Resources Project (WAT) has been addressing water quality aspects of rivers and lakes for the past fifteen years. Water quality management of highly polluted rivers in Central and Eastern Europe (CEE) is one of the major concerns of WAT. The present paper is based on a comprehensive investigation performed to identify management options for the Nitra River basin in Slovakia. The Nitra river serves as a case study in the frame of a joint research effort with the Water Research Institute (VUVH, Bratislava) and the Váh River Basin Authority. The aim was to identify the most appropriate wastewater treatment strategies for the municipal wastewater discharges in the Nitra River basin. A broader objective was to develop models, methodologies and policy conclusions of wider applicability and interest in the CEE region. This is one of a series of papers that describe the various components of the study.

Abstract

The Nitra is one of the most polluted rivers in Slovakia due to numerous municipal and industrial discharges, and the low level of waste water treatment. The ongoing economic transition and lack of financial resources for environmental management calls for the development of short-run least-cost policies on the basis of ambient standards (or a combination of ambient and effluent ones). A water quality control policy model was developed which incorporates dissolved oxygen simulation models, municipal wastewater treatment alternatives, an optimization model based on dynamic programming, a data base, and a graphical user interface. Least-cost policies to achieve various water quality goals were developed and compared to effluent standard based strategies (including that deriving from the application of the "best available technology"). The role of industrial emissions was demonstrated in a sensitivity fashion, while the influence of parameter uncertainty on the developed policies was analyzed in a multiobjective framework. The study shows that significant cost savings are possible in comparison to uniform, effluent standard policies. They also suggest that a long-term strategy should be realized on the basis of a sequence of properly phased least-cost policies corresponding to ambient standards to be tightened gradually.

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WATER QUALITY MANAGEMENT OF THE NITRA RIVER BASIN (SLOVAKIA): EVALUATION OF VARIOUS CONTROL STRATEGIES

L. Somlyódy, M. Kularathna, and I. Masliev

1 INTRODUCTION

The Nitra River is a tributary of the Váh which enters the Danube downstream to Bratislava where the Danube forms the border between Slovakia and Hungary. The length of the river is about 170 km. The mean streamflow near the mouth is 24 m^3/s, while a typical August low flow condition is characterized by not more than 3 m^3/s. The watershed area is 5100 square km and 650 000 inhabitants live there (about 40% of them live in rural areas). The region is highly industrialized. Metallurgical works, meat and sugar beat factories, tanneries, chemical plants etc. can be found at various locations. Level of water re-cycling and industrial wastewater treatment is low. Agricultural non-point source pollution is negligible. Sewage networks and wastewater treatment plants can be found in eleven larger municipalities (with a total population of 350 000). The level of public water supply as well as sewerage in these municipalities is close to or above 90%. (however in rural settlements, only a small portion of the population is supplied by potable water, and wastewater treatment is practically non-existant). The level of wastewater treatment is below 50%. Treatment plants, most of which based on the activated sludge process without nitrification, are about 20 years old. Several plants are overloaded by 100% or more, leading to biochemical oxygen demand (BOD5, which will be denoted hereinafter as BOD) removal rates between 60% and 70%. The excess flow above the design capacity is generally given only primary treatment. A map of the watershed illustrating the levels of municipal wastewater treatment is presented in Fig.1. The level of industrial wastewater treatment is shown in Fig. 2.

The water quality of the river is one of the poorest in Slovakia. Parameters of the oxygen regime and other chemical components specify Class IV-V quality, according to the existing classification system (V indicates the worst quality). For instance, BOD values can exceed 30 mg/l, and dissolved oxygen (DO) is sometimes below 1-2mg/l. Total phosphorus (TP) can reach 2 mg/l, while ammonia nitrogen (NH4-N) can be around 7 mg/l. High algae biomass levels can be observed at the downstream stretch of the river (a-chlorophyll values of 150 mg/m^3 are not rare), where travel time is sufficiently long.

Municipalities contribute about 70% of the total emission of traditional pollutants (organic material, phosphorus and nitrogen) of which 30-50% is due to industrial discharges (see Figs. 1 and 2). Industrial contamination is characterized, among others, by high arsenic concentrations (the origin of which is yet unknown) and high conductivity.

At present, the primary utilization of the Nitra River is waste disposal (as apparent from what was outlined above) and water abstraction for industrial and irrigation purposes. For the latter purpose, small dams have been built, mostly at the downstream part of the river.

The economy of the country and the region is in a strong transition. In spite of increasing water prices, domestic water consumption has not yet been reduced significantly. The same statement also applies for industries connected to municipal wastewater treatment plants (WWTPs).

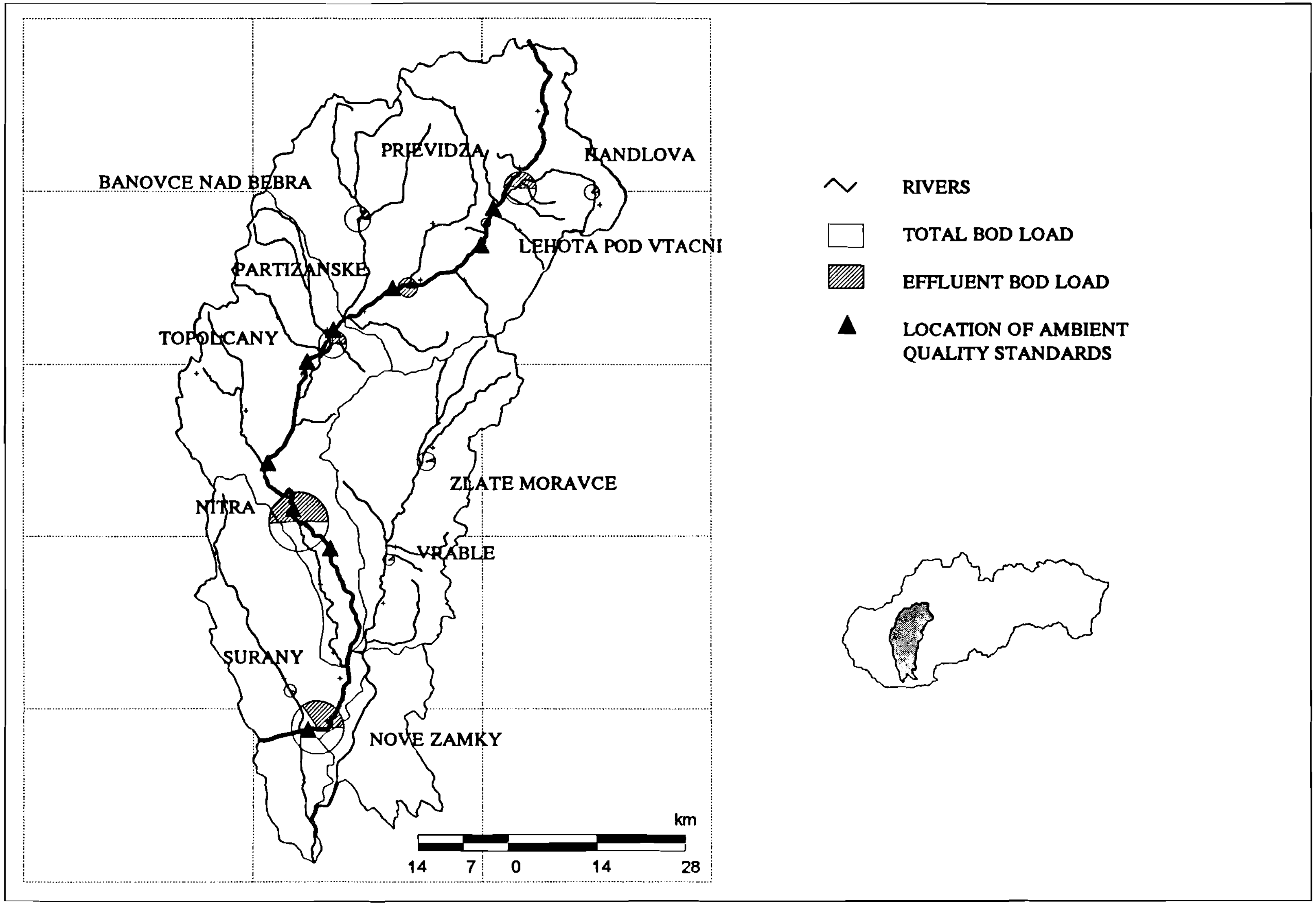

Fig.1 Municipal wastewater treatment in the Nitra basin

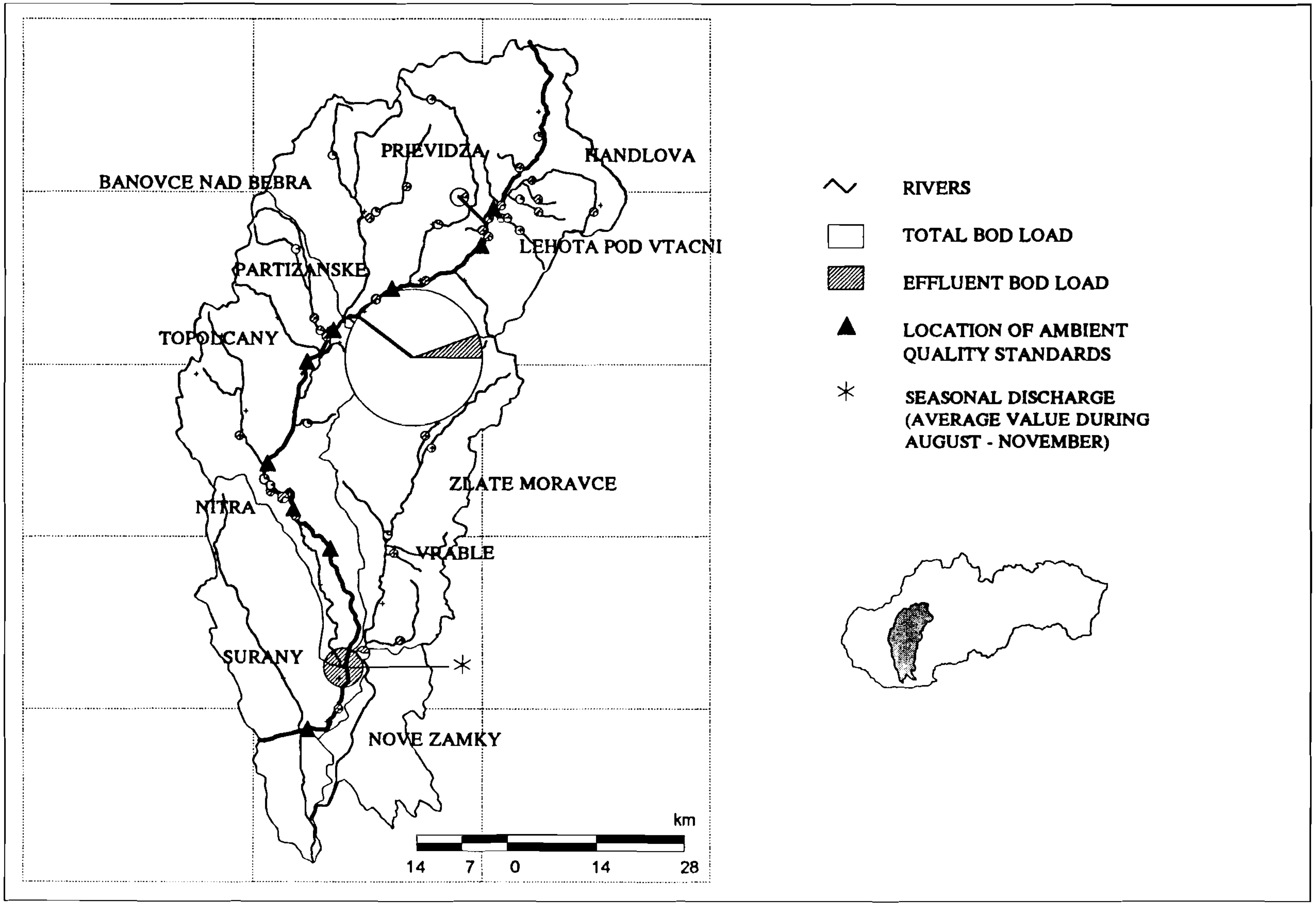

Fig.2 Industrial wastewater treatment in the Nitra basin

Thus, the flow and load of WWTPs have been reduced only slightly during the past three years. In contrary, larger industrial plants outside of urban areas went through significant changes. A large portion of them were fully shut down. Some others show about 50% discharge reduction and only a few operate in an unchanged fashion. In the short run, this trend is likely to be continued, while longer term alterations are hard to visualize.

A new effluent standard system was set in 1992 by the earlier Czech and Slovak Republics as a basis for future legislation. This system leaves open the possibilities to also incorporate ambient water quality criteria. The procedure realistically distinguished two stages - before and after 2004. For a 100 000 population-equivalents or larger municipality the standards are as follows: BOD=30 mg/l, NH4-N=10 mg/l and TP=3 mg/l. The corresponding limits are slightly more stringent as of 2005.

Under the present economic conditions, financial resources are not available for major environmental investments. In spite of that, the consequences of specifying the above effluent criteria, which demand major investments, have not been analyzed. Thus, the issue of developing a least-cost policy (based on ambient quality criteria) on the short run was raised.

The development and discussion of this strategy (together with effluent standards based ones and mixed ones) is the major objective here, specifically for the Nitra watershed. The work to be outlined is an element of a comprehensive, policy oriented research program with the involvement of the Water Research Center (Bratislava) and the Váh River Basin Authority (for details the reader is referred to Somlyódy et al., 1993). The other goals and components of the study include: a survey of various emissions, the development of wastewater treatment alternatives and upgrading strategies, longitudinal water quality profile measurements and their evaluation (see Masliev and Somlyódy, 1993), and the application of various water quality simulation models (from that of Streeter and Phelps to QUAL2e; see Breithaupt and Somlyódy (1993) for the latter) to be used for policy purposes.

The paper is organized as follows. A brief literature review on water quality management models is presented first. The approach based on dynamic programming (DP) is described subsequently. DP is an attractive methodology which can exploit the sequential characteristics of river basins (downstream control actions do not influence water quality upstream). It can also handle the discrete decision variables that represent different treatment levels. Furthermore, it is generic in that both linear and non-linear water quality models expressing the relation between emissions and ambient water quality can be incorporated. The results presented herein include different control strategies developed with a focus on municipal emissions and oxygen household. A discussion of several policies is also given. The role of industrial emissions is demonstrated in a sensitivity fashion. Finally, the influence of parameter uncertainty (of the traditional dissolved oxygen model employed at this stage) on policies developed is analyzed in a multiobjective framework.

2 AN OVERVIEW ON WATER QUALITY MANAGEMENT MODELS

In general, simulation models, optimization models, or a combination of these two have been employed to determine the "optimum" or the satisfying set of wastewater treatment alternatives for river basins. These methods assist a decision maker in reaching a solution which is optimal (or near-optimal) with respect to some predefined goals. Often there are economic and water quality goals to be considered. The most obvious economic goals are the cost-minimization and proper distribution of costs (in space and time). Water quality goals are usually expressed by quality standards for the receiving waters and/or for the wastewater effluents.

In order to select an "optimal" set of wastewater treatment alternatives by simulation, a river model has to estimate the consequences of a variety of feasible alternatives. The number of feasible combinations of treatment alternatives in a river basin increases rapidly as the number of treatment plants and treatment levels increases. Consequently, a large number of simulations are required to select an "optimal" set of alternatives. In order to incorporate the uncertainty of various inputs and parameters, many more simulations in a Monte-Carlo fashion would be needed. This will increase the required number of simulations to an impractical level. Nevertheless simulation offers the possibility for a more detailed representation of a river system. Simulation models in water quality management include those presented by Warren and Bewtra (1974), Yeh (1973), Orlob (1982), and Thomann and Mueller (1987). The role of BOD dischargers in the Nitra river basin was analyzed using a simulation model by Koivusalo et al. (1992). A state-of-the-art review of water quality simulation models, including a discussion on decision support systems, can be found in Somlyódy and Varis (1992).

When considering the optimization techniques, linear programming (LP) has been one of the most widely used techniques in water quality management. Its wide popularity is partly due to the availability of general-purpose LP packages. LP solves a special type of problem in which all relations among the variables are linear. This requirement should be fulfilled by the objective function as well as by the constraints of the model formulation. It is however possible to linearize nonlinear problems under certain assumptions, so that they can be solved by linear programming. The typical objective function for a LP application would be to minimize the cost that satisfies all the water quality standards in the river system. Convex cost functions are also a requirement for the successful application of LP. These requirements together with the integer properties of decisions of a water quality management problem dictate that the problem often has to be reformulated in order to be solvable by LP. Such reformulations may lead to suboptimal solutions. LP applications for water quality management problems have been presented by many researchers, including Loucks et al. (1967), ReVelle et al. (1968), Bishop et al. (1974), Biswas (1981), and Burn and Lence (1992).

Linear mixed-integer programming (MIP) is also appropriate to solve this problem, as the feasible decisions comprise of a discrete set of wastewater treatment alternatives. It has an advantage that the problem need not be reformulated as a continuous one. However MIP imposes severe limitations on the size of the problem due to the associated high computational load. Hughes (1971), and Loucks et al. (1981) have presented MIP formulations for water quality management problems.

The wastewater treatment cost functions as well as the "transformation functions" that relate waste discharge to the river water quality are generally nonlinear. Therefore a nonlinear programming model would represent the reality more accurately than a linear model. Nevertheless they have been less popular in water quality management applications. This is due to the complexity of such approaches and their large computational requirement in comparison to the other methods. Furthermore they cannot easily incorporate uncertainty in the system. Hwang et al. (1973), Bayer (1974), and Pratishthananda and Bishop (1977) have applied nonlinear programming for river water quality management.

Some of the optimization techniques permit the explicit incorporation of uncertainty into the model formulation. Water quality problems deserve such considerations, as they are characterized by an array of uncertainties. Several different methods are available to handle uncertainty in a LP formulation. Stochastic linear programming is one such method, which results in a large increase in the number of variables and constraints. This is a result of incorporating various alternative scenarios into the model formulation explicitly. An alternative approach is to use chance constraints. It reformulates the stochastic problem into a corresponding deterministic one by defining constraints based on certain reliability levels. This is considered as a pessimistic approach. Also, the selection of the reliability values (which can even be a part of the optimization problem) is not a straightforward task.

Modelling approaches that combine simulation and optimization have also been reported, mostly for linear, steady state problems where water quality models can be replaced by the so-called transmission coefficients (see Burn, 1989). Such models very often perform a decomposition of the problem. As an example, Monte-Carlo simulation of the system followed by optimization for each generated scenario may be helpful in deriving probabilistic conclusions on the control actions. Such approaches have been documented by deLucia and McBain (1981), and Burn (1989). An approach based on multiple scenarios (reflecting alternative hydrologic, meteorologic and pollutant loading conditions) was presented by Burn and Lence (1992). Somlyódy and Wets (1988) described a linearized expectation-variance approach together with a quadratic stochastic optimization for a lake problem.

Sobel (1965) proposed the use of stochastic linear programming for the optimizations in water quality problems. Deininger (1965) applied a chance constrained LP method using an approximation of the Streeter-Phelps equations. Lohani and Thanh (1979) incorporated the random nature of streamflow into the waste load allocation problem by using chance constraints. Fujiwara et al. (1988) presented a study on stochastic water quality management. A stochastic dynamic programming approach was presented by Cardwell and Ellis (1993).

Applications of dynamic programming (DP) for water quality management problems have been reported by Newsome (1972), Hahn and Cembrowicz (1981), and Cardwell and Ellis (1993). A discussion about the DP approach and the reasons for its selection in the present study are presented below. For a more detailed description of the state-of-the-art of water quality management models, the reader is referred to Kularathna and Somlyódy (1993).

3 THE APPROACH BASED ON DYNAMIC PROGRAMMING (DP)

Dynamic programming (Bellman, 1957) is an optimization method for a multistage decision problem. It decomposes a problem with a sequence of decisions into a sequence of sub-problems each having one or a reduced number of decisions. In fact, these subproblems are solved recursively, by considering the sub-optimal solution(s) of one subproblem as input(s) to the subsequent subproblem. The selection of optimal wastewater treatment alternatives in a river basin is a sequential decision problem. It involves sequential decisions in space as well as in time. Spatially, the decisions are to be taken for a series of locations in a river basin. Due to the downstream-only propagation of pollutants in a river system, the water quality at a particular location in a river is fully determined by the water quality at the immediate upstream discharge/control point (or by several discharge/control points in the special case where the considered location is below a confluence). Similarly, in planning the investments over the planning horizon, the decisions are to be made at a sequence of points in time. The decisions made at one time point directly affect those to be made at the next time step. These special sequential attributes of the problem make it very suitable to be solved by a dynamic programming approach.

Model linearity is not a requirement for DP. This opens up the possibilities to incorporate complex non-linear water quality models within the optimization process. Most of the other optimization techniques would require significantly simplified forms of such models. Furthermore, it can incorporate stochastic features as well. In discrete DP, constraints that reduce the state or decision space (e.g. prespecified water quality standards) are advantageous because they reduce the computations. Such constraints cause additional computational burden in other optimization techniques.

The main limitation of DP is due to the rapid increase of the computational load as the number of state variables increases. This is appropriately known as the "curse of dimensionality" of dynamic programming. Separability of the objective function is a requirement of a problem to be solved by DP. However this requirement can be relaxed by alternative formulations (by including additional state variables) which increase the size of the problem.

3.1 DP Formulation

The river system was subdivided into a number of reaches that were further divided into "stages". The river network has been defined by the interconnection of different reaches. A "stage" was considered as a part of the river from a point immediately upstream of a "point of action" to a point immediately upstream of the next "point of action" downstream. A "point of action" can be: a wastewater discharge, an abstraction point, a measurement point, a point with a prespecified water quality standard, a weir or one of the artificial points introduced to maintain a generalized computational procedure.

The computations of DP were started from the most upstream point of the river system. Using Bellman (1957)'s principle of optimality, the DP calculations were performed stage by stage, proceeding towards the most downstream point of the system. A description of the DP computations is given below, for a simplified case of a river without any tributaries. An extended form of this procedure was employed to analyze the Nitra system comprising of several tributaries. The optimization problem for the most upstream stage can be expressed as:

$$f_1(Q_1) = \min_{\eta_1} C_1(\eta_1)$$
$$Q_1 = T_1(\eta_1, Q_0)$$

where, $f_n(Q_n)$ (n=1 for the most upstream stage) is the cumulative optimal cost required to achieve the allowable water quality state Q_n at the end point of stage n. $C_n(\eta_n)$ is the treatment cost required (which is zero if there is no wastewater discharge or if no treatment is to be made at stage n) to achieve an efficiency of η_n at stage n. This efficiency η_n (the removal rate) is the decision variable at stage n of the problem. For the most upstream stage, Q_0 indicates the headwater quality. $T_n(\ )$ represents the transfer function at stage n, and is a water quality simulation model. The treatment cost considered in this study was the total annual cost (TAC); which is comprised of the contributions of investment cost (IC) and the operation, maintenance, and replacement cost (OMRC).

The efficiency of a particular treatment alternative is characterized by the influent and effluent quality. For each WWTP, a set of feasible treatment alternatives were developed. Each alternative is represented by its IC, OMRC, and the effluent concentrations (see below). The optimization task is to make a (0,1) decision with regard to all feasible treatment alternatives at each of the WWTPs. In fact, these decisions should sum-up to one at each WWTP location.

For the subsequent stages (from n=2 upto the most downstream stage), the recursive relation takes the following form:

$$f_n(Q_n) = \min_{\eta_n,\, Q_{n-1}} [C_n(\eta_n) + f_{n-1}(Q_{n-1})]$$
$$Q_n = T_n(\eta_n, Q_{n-1})$$

The standard approach in using discrete DP is to discretize the state space and represent each discrete quality state by a representative point. This representative point is usually a fixed one (center of the discrete state, for example). Some rounding off is often required at this point, because the simulated water quality may not exactly coincide with the central point of the quality state. However, the representative point used in the current approach is the simulated quality state which corresponds to the best objective function value within the current state. This allows a correct representation of the river water quality profile (for the selected control actions), as the rounding off of quality figures are not involved.

During computation, the current decision (treatment alternative), the previous state (at the previous stage) and the cumulative cost are recorded for each allowable water quality state at each stage. Having reached the most downstream point to be considered, the optimal solution can therefore be traced-back upstream.

The DP approach can be extended to handle the scheduling problem, although it would result in a high computational load. For this, it would be necessary to determine a set of acceptable strategies for each time step separately. The allowable transitions of each strategy to those in the next time step need to be identified. Resulting problem is a standard discrete dynamic programming problem, of selecting the optimal plan characterized by one treatment strategy at each time step. It is to be noted, however, that incorporation of uncertain scenarios would further increase the amount of computation.

4 APPLICATION TO THE NITRA RIVER BASIN

4.1 Water Quality Simulation Models

The application of a number of water quality models describing the household of oxygen and nutrients is under way for the Nitra River. All of them use a general, steady state river network hydraulic model supplying flow and geometric data. For policy purposes, the traditional Streeter-Phelps model and its extensions were tested (Masliev and Somlyódy, 1993). Different model versions were calibrated to the longitudinal water quality observation performed in August 1992. At the end, the usage of the three-state variable, linear version was decided upon, incorporating carbonaceous and nitrogenous biochemical oxygen demand (CBOD and NBOD), as well as DO. This model is simple, but still offers some information on ammonia (which can be obtained by stochiometric conversion). Parameters considered were the decay rate for CBOD and NBOD, respectively (assumed to be identical, K_d) and the reaeration coefficient K_r (the settling rate of particulate CBOD and the sediment oxygen demand were set to zero within the management model solely for the sake of simplicity).

The estimated mean decay rate (K_d) was around 0.8 l/d, which is higher than the recommendation of the literature. The explanation is twofold. First, the presence of partial biological waste water treatment only and, second, the small water depth. The mean reaeration coefficient (varying longitudinally) was about 2.0 l/d, leading to a realistic K_r/K_d ratio. On one hand, a sensitivity analysis showed the dominating role of the reaeration coefficient on the DO profile. On the other hand, the statistical distributions of parameters were available from a Monte Carlo parameter estimation performed by Masliev and Somlyódy (1993). The coefficient of variation was about 15% for K_r, and this value will be employed for the deterministic multiobjective analysis. The ongoing research activities that are not described here include: stochastic extension of the present approach, the incorporation of more complex, non-linear water quality models, scheduling of the implementation of treatment strategies, and the systematic evaluation of various management model formulations in a broad sensitivity study framework.

4.2 Design Condition

Results of the August 1992 experiment (Masliev et al., 1993) performed under critically low flow conditions served as the basis for the (single) design scenario (including the derivation of parameters of the DO model). Background pollution of the tributaries and the temperature profile (showing an increase downstream from 19^oC to about 25^oC) were obtained from the observations. For the generation of design emissions for municipalities, annual average values available for the period 1990-1992 were used in addition to measured ones. Industrial emissions were kept at their 1990 level, mainly as a worst-case scenario and to demonstrate the role of changing industrial emissions (as observed recently) by a sensitivity study. The optimal policies to be developed will focus on municipal emissions. Industry will be considered without analyzing the costs, due to lack of reliable information on industrial discharge control measures.

4.3 Municipal Waste Water Treatment Alternatives

For all the WWTPs in the region, analyses were performed to develop feasible treatment alternatives. Each alternative is characterized by the flow, the effluent quality expressed by DO, BOD, SS, NH4-N, nitrate nitrogen (NO3-N) and TP, as well as the estimated costs, IC, OMRC and TAC. The alternatives were obtained by considering existing units (primary sedimentation tank, aeration basin, final clarifier and sludge processing), their various upgrading possibilities, and the design of new treatment plants. The average project life time was assumed to be 20 years (longer for new plants and shorter for upgraded ones). In addition, a 10% interest rate was employed to obtain total annual costs.

Feasible treatment alternatives at each WWTP were assigned numbers (treatment levels) starting from 0 (no-treatment was indicated by Level 0, while the most expensive solution by the highest level, see below). The number of levels ranged between four and ten, depending on the actual situation and technological calculations. The basis of deriving alternatives were as follows:

- Levels 0 corresponds to no treatment, while Level 1 assumes the operation of existing facilities as it is the case nowadays. The demolition of existing WWTPs and the construction of new, advanced plants characterized approximately by BOD $\leq$ 15 mg/l, TN $\leq$ 10 mg/l and TP $\leq$ 1 mg/l effluent quality (corresponding to the most stringent recommendations of the European

Community) was considered as the highest "level" alternative (called "best available technology", BAT).

• The increase of the number and size of units (without changing the type of processes) to compensate overloads (depending on technological calculations) resulted in other alternatives.

• Upgrading by adding a low dosage of chemicals before the primary clarifier was considered as an attractive, cost-effective alternative for most of the WWTPs (which leads to a combined chemical-biological treatment, see Henze and Ødegaard, 1994). As shown in the literature, the surface overflow rate and BOD removal of primary sedimentation basins can be practically doubled (Morrissey and Harlemann, 1990). Thus, the capacity of overloaded biological treatment plants can be significantly extended (depending on the capacity of the final clarifier and the sludge line). Jar tests performed at various plants in the Nitra Basin justified the applicability of this method (Murcott and Harlemann, 1994).

• The combination of the above upgrading options is also possible. Similarly, the re-shaping of existing plants and the construction of future WWTPs (which can be based on different technological principles, see Henze and Ødegaard, 1994) can also combine different options (which lead to additional levels). For instance the flow of an existing, overloaded facility can be reduced (even to a nitrification operation mode resulting in a "downgrading") to improve the performance. This would require the construction of a smaller new plant than under the condition of pulling down the old one.

Additional alternatives can correspond to present or future effluent standards in Slovakia, and thus, a comparison to other strategies is a straightforward task. Examples for this can be given with regard to the two largest WWTPs in the basin: Nitra and Nove Zamky. The design capacity of both of them were slightly above 10 000 m^3/d, while the present flow is close to 30 000 m^3/d. Table 1 displays the feasible treatment alternatives and approximate cost estimates for Nove Zamky treatment plant. The BAT would cost more than 20 million USD each. The "cheapest", still effective upgrading which improves the effluent BOD value from about 60-80 mg/l to approximately 20 mg/l would cost about 3-6 million USD, while a combination of a "downgraded" old plant and a new one would amount to 11-14 million USD. Further details on cost-effective treatment strategies and applications to the Nitra River basin, we refer to Somlyódy (1993) and Somlyódy et al (1993).

Table 1 Characteristics of the Feasible Treatment Alternatives for Nove Zamky WWTP

Alternative	Cost (10^6 USD)		Effluent Concentrations (mg/l)			
	IC	OMRC	BOD	TP	NH4-N	NO3-N
0	0	0	240	11	48	0
1	0	1.0	60	9	40	0
2	3	1.2	15	7	4	30
3	2	1.4	20	1.5	34	0
4	5	1.6	15	1.0	4	30
5	6	1.9	15	1.0	2	17
6	13	1.2	15	7	4	30
7	11	1.6	15	1	4	26
8	14	1.9	15	1	0	19
9	21	2.0	10	0.8	0	10

4.4 Development of Different Water Quality Control Policies

The analysis outlined above led to a set of tables for each treatment plant. As displayed in Table 1, the lines of the tables represent alternatives or levels characterized by effluent water quality and costs. To each line a (0,1) decision variable is associated which guaranties also the linkage to dynamic programming (see above).

A summary of the results obtained with the water quality control policy model using dynamic programming is given in Table 2. It incorporates a number of strategies, together with IC and OMRC costs, water quality extremes for DO, BOD, and NH4-N (outside of locations where ambient criteria were set as constraints), as well as the sum of treatment levels (defined above)for all the sites. The first five strategies are based on effluent standards, and thus, both ambient water quality and costs are direct consequences (no optimization is needed). The second block [(6)-(9)] represents least-cost strategies defining ambient criterion alone for DO. The next two lines illustrate the consequences of having additional constraints for BOD and NH4-N, respectively. (corresponding to quality Class III). Policies (12) and (13) are mixed ones; ambient and effluent criteria are used jointly. Finally, the last block shows the influence of industrial emissions. The first two lines were obtained with the aid of simulation and they show the impact of uniformly reduced industrial emissions on water quality. In contrary, the last two lines illustrate least-cost strategies under the reduction of industrial loads by 50% and 75%, respectively. Longitudinal profiles of DO, BOD, and NH4-N corresponding to each alternative of Table 2 are presented in Figs. 3-5 respectively. It is to be noted that the steep changes in the concentrations displayed in these figures result from one or more pollutant discharges, or, highly polluted tributaries. Conclusions from the results can be drawn as follows.

TABLE 2 A Summary of Different Control Policies

No:	Policy*	IC [mil. USD]	OMRC [mil. USD]	DO min [mg/l]	BOD max [mg/l]	NH4-N max [mg/l]	Sum of treatment levels
1	No treatment	0.0	0.0	0.1	34.6	7.9	0
2	Current treatment	0.0	5.7	2.3	30.6	7.7	10
3	Eff: BOD≤30, NH4-N≤10, TP≤3	32.1	8.4	5.4	11.3	2.3	35
4	Eff: BOD≤25, NH4-N ≤5, TP≤1.5	35.2	8.7	5.4	11.3	2.3	37
5	Best available technology	95.5	11.1	5.7	11.1	2.1	53
6	DO≥3	3.2	3.8	3.0	18.7	4.2	8
7	DO≥4	4.0	5.6	4.2	13.8	4.1	11
8	DO≥5	15.0	6.2	5.0	13.8	3.2	18
9	DO≥6**	33.4	7.7	5.6	11.3	2.2	26
10	DO≥5, BOD ≤10	20.0	7.0	5.0	11.2	3.6	20
11	DO≥5, BOD≤10, NH4-N ≤2	29.5	7.1	5.6	11.3	2.2	24
12	DO≥5, BOD≤10, Eff: NH4-N≤10	31.0	7.9	5.6	11.1	2.2	29
13	DO≥5, BOD≤10, Eff: NH4-N≤10, TP≤3	33.6	8.5	5.5	11.3	2.3	36
14	DO≥5, BOD ≤ 10, simul: with 50% ind.	20.0	7.0	6.0	8.5	3.5	20
15	DO≥5, BOD ≤ 10, simul: with 25% ind.	29.5	7.1	5.5	8.2	3.4	20
16	DO≥5, BOD ≤ 10, 50% ind.	7.2	6.2	5.1	9.7	3.6	14
17	DO≥5, BOD ≤ 10, 25% ind.	6.2	5.7	5.1	11.4	3.8	12

* Standards are expressed in mg/l

** Constraint violated at one of the standard points

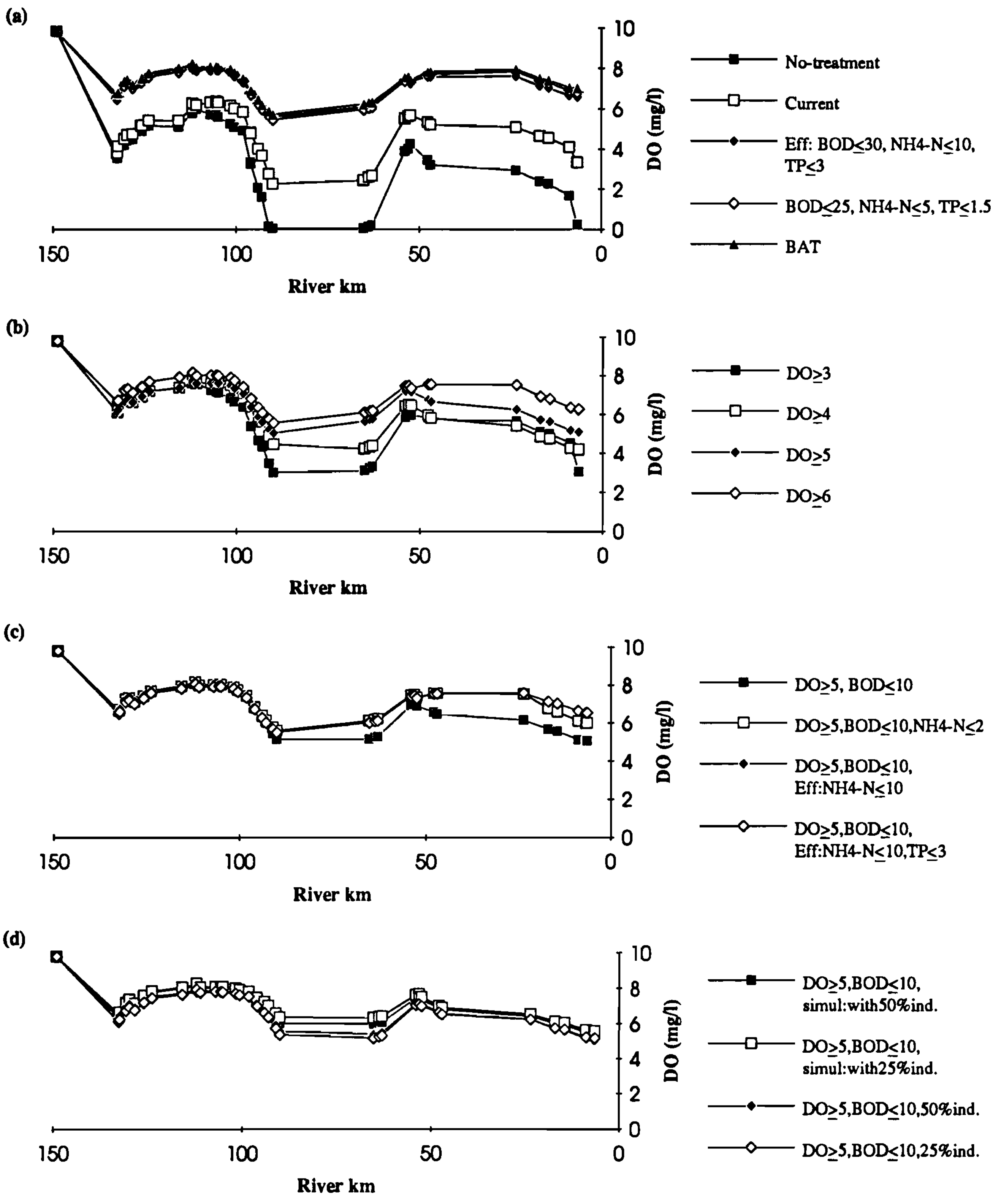

Fig.3 Longitudinal profiles of DO concentration
(a) Policies 1,2,3,4,5; (b) Policies 6,7,8, 9; (c) Policies 10,11,12,13; (d) Policies 14,15,16,17

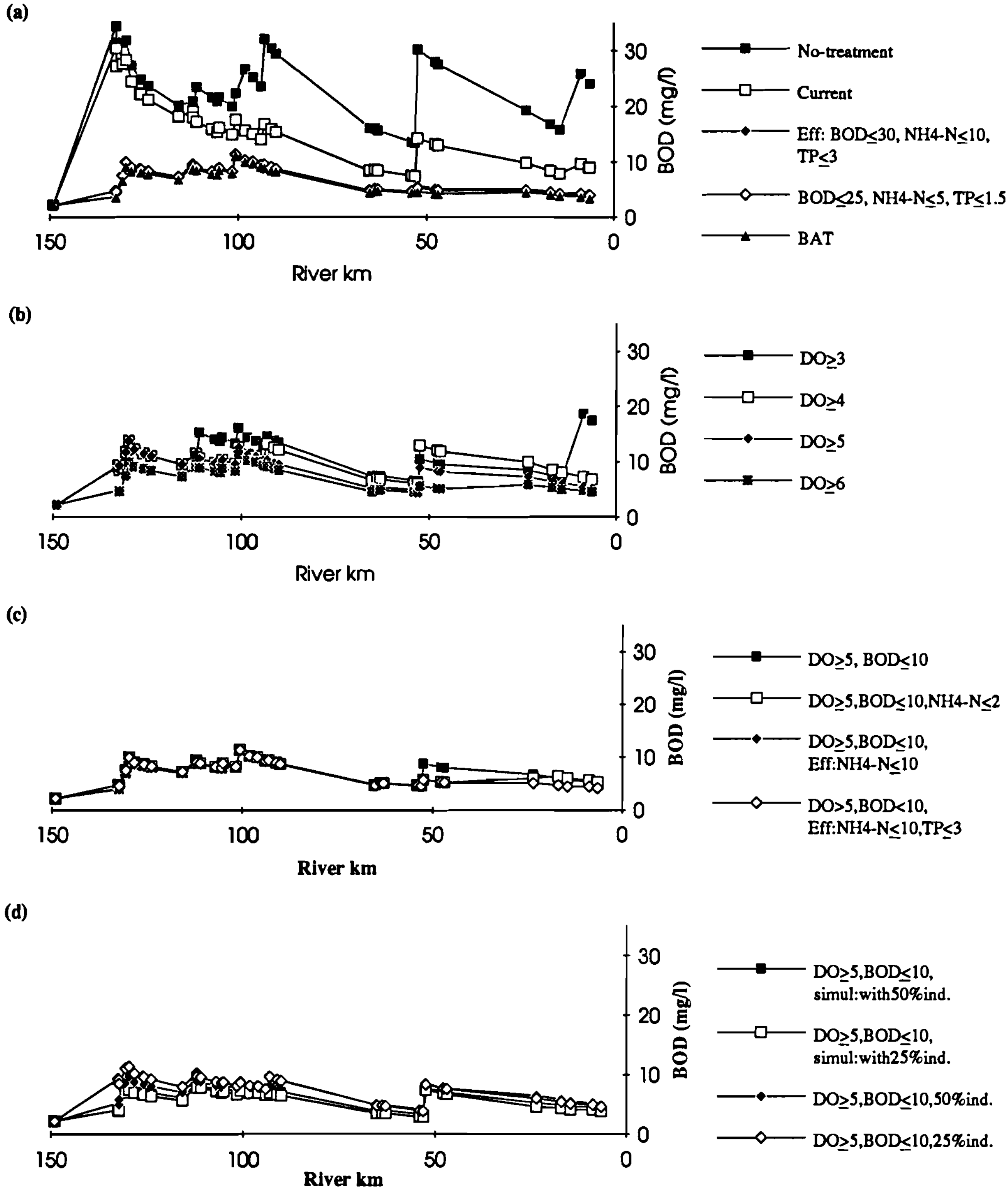

Fig. 4 Longitudinal profiles of BOD concentration
(a) Policies 1,2,3,4,5; (b) Policies 6,7,8, 9; (c) Policies 10,11,12,13; (d) Policies 14,15,16,17

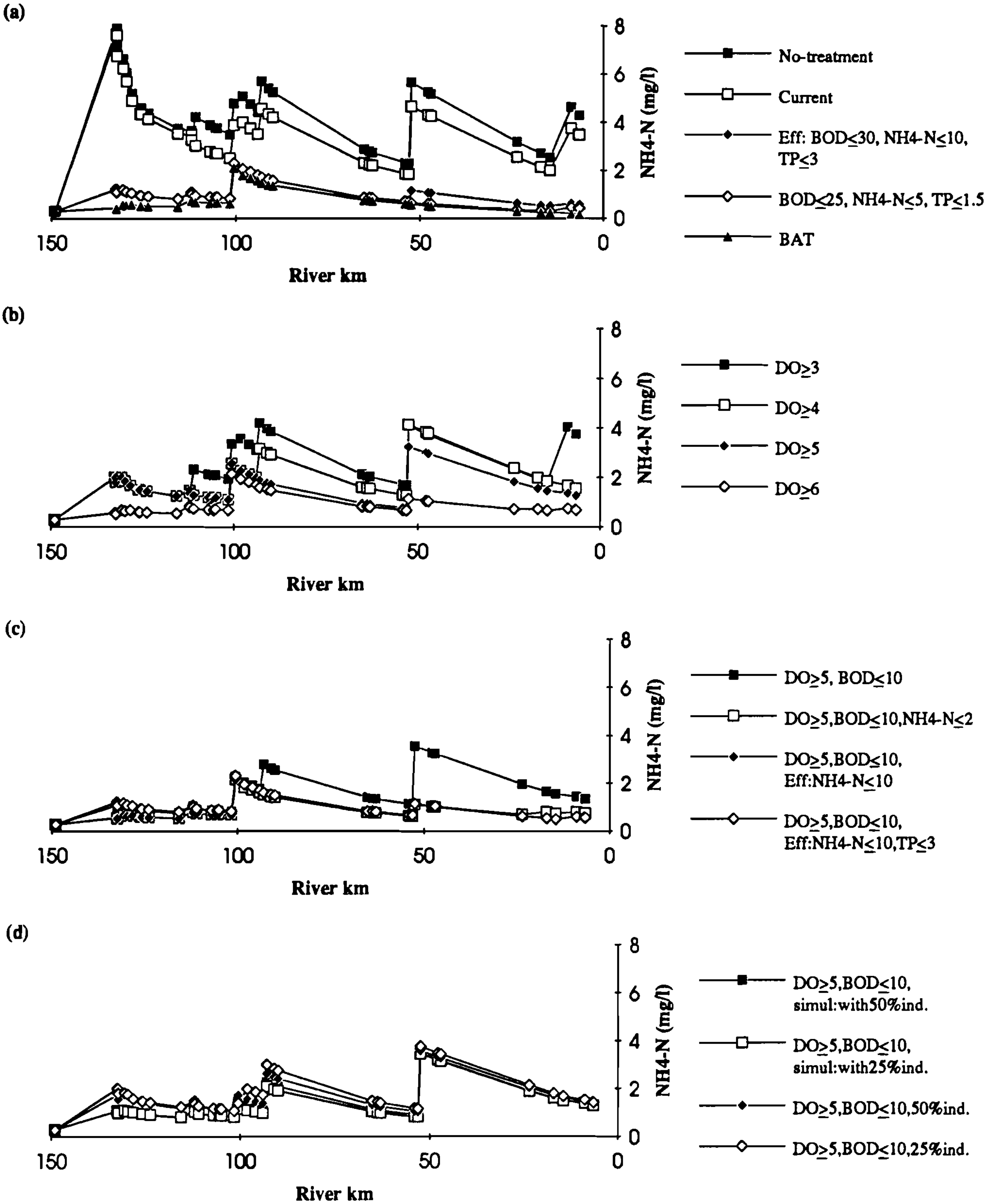

Fig. 5 Longitudinal profiles of NH4-N concentration
(a) Policies 1,2,3,4,5; (b) Policies 6,7,8,9; (c) Policies 10,11,12,13; (d) Policies 14,15,16,17

• The range of water quality changes due to control is relatively narrow (e.g. between 11 and 35 mg/l for BOD) since only municipal emissions are incorporated into the optimization model. The combination of BAT for municipalities and 75% uniform industrial discharge reduction leads to a maximum BOD concentration (BOD_{max}) of about 8 mg/l and a minimum DO concentration (DO_{min}) close to 7 mg/l (the average saturation value was around 8 mg/l). This result illustrates the role of industry and "background" pollution (considered "uncontrollable") alike. The optimal budget allocation would be different from the present one if all the discharges of differing origins were accounted for. In contrast to water quality, the domain of meaningful expenditures is extremely broad, between 3 and 95 million USD. It indicates significant saving possibilities. The longitudinal profiles of ambient DO concentration corresponding to the strategies (2), (3), (5), (6) and (8) are presented in Fig. 6. It can be observed that the strategies (3), (5) and (8) give rather identical DO profiles, although their investment cost requirements are very different (32, 95 and 15 million USD respectively).

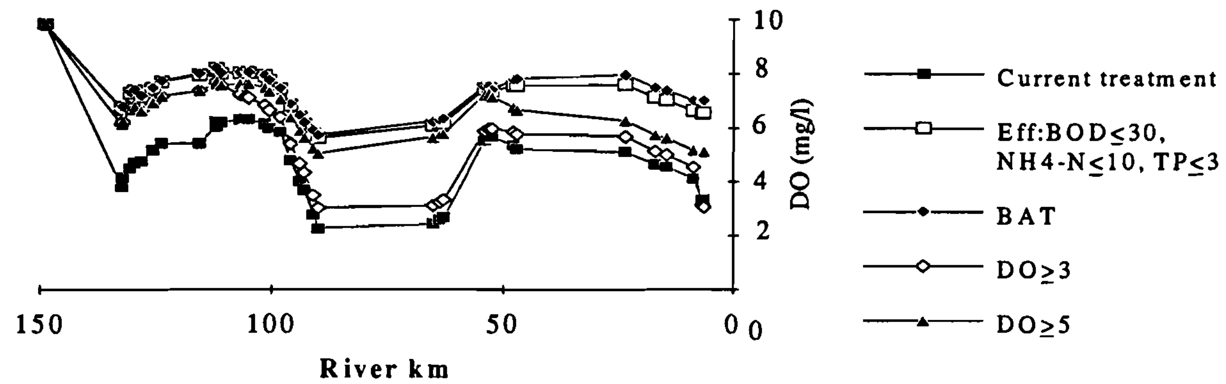

Fig. 6 Longitudinal profiles of DO concentration for the policies 2, 3, 5, 6, and 8

• Four clusters of policies can be identified from the table. 3-4 million USD is the smallest investment which leads to measurable improvement in DO (and also in the other two components). The second cluster is formed by policies (8) and (10) in the 15-20 million USD capital cost domain (DO≥5 and BOD=11-14 mg/l). The third group comprises strategies (3), (4), (9), and (11)-(13). The investment cost is 30-35 million USD and the water quality further improves (DO≥5 mg/l, BOD~11mg/l, and NH4-N~2 mg/l roughly corresponding to Class III water). Nevertheless, the marginal cost of this improvement is rather high. It is worthy to note that policies (3) and (4), corresponding to present and future effluent standards in Slovakia are identical from the viewpoint of ambient quality, in spite of the difference in costs. Finally, the last cluster is represented by the construction of new BAT treatment plants. Short term benefits are absent; the capital cost is three times higher than for the previous set, but water quality is practically unchanged (Class III). However, BAT reduces the NH4-N profile more pronouncedly (Fig. 5a) than the DO and BOD profiles, due to the associated denitrification process.

• The economic implications of setting standards is strong, non-linear and somewhat surprising. For instance, the improvement of DO_{min} from 4 mg/l to 5 mg/l requires 11 million USD investment. Similarly, the price of specifying BOD as a criterion in addition to DO (not necessarily justified professionally) is 5 million increase in the cost (see Table 2, lines (8) and (11)). The incorporation of NH4-N has an even stronger influence. The cost implications of setting DO and NH4-N standards are summarized in Fig. 7, considering three different scenarios for the industrial discharges based on their 1990 values (no reduction, 50% reduction, and 75% reduction, of 1990 discharges). It is to be noted that the peculiar low IC associated with the standards DO≥6, NH4–N≤2 in Fig. 7(e) is offset by a higher OMRC cost; indicating a higher amount of total annual cost.

• The reason for the above nonlinear behavior (of the economic implications of setting standards) is the change in the sum of treatment levels and in the spatial configuration. Treatment levels corresponding to each strategy in Table 2, for each location, are displayed in Table 3. As can be seen from Tables 2, although there is hardly any difference between strategies (2) and (7) in the total number of treatment levels, the latter non-uniform policy (see Table 3) is much more efficient. The transition from policy (7) to (8) is achieved primarily by investing in the upgrading of two larger WWTPs, Topolcany and Nitra (Fig. 1), respectively. In contrast to effluent standard based strategies, all the least-cost policies are strongly non-uniform, reflecting longitudinal changes of water quality and the local cost-effectiveness of individual control measures. Mixed policies (e.g. (12) and (13)) show more uniformities.

• Treatment technologies remove different pollutants simultaneously, but to different extents. The multiple pollutant nature of the problem and the alternative treatment methods is well-reflected by the table. Total phosphorus (TP) plays a less important role in this respect. The influent TP concentration is rather small (due to high water consumption and infiltration to the sewerage), and its reduction to levels set by effluent standards can be realized relatively easily.

Table 3 Treatment Levels Corresponding to The Strategies in Table 2

Policy*	Treatment Plant**										
	Ha	Le	Pr	Pa	Ba	To	Ni	Zl	Vr	Su	No
No treatment	0	0	0	0	0	0	0	0	0	0	0
Current treatment	1	1	0	1	1	1	1	1	1	1	1
Eff: BOD≤30, NH4-N≤10, TP≤3	2	2	5	3	3	6	5	2	1	2	4
Eff: BOD≤25, NH4-N ≤5, TP≤1.5	2	2	5	3	3	6	6	2	2	2	4
Best available technology	4	3	4	3	5	8	7	4	3	3	9
DO≥3	0	0	2	0	1	2	2	0	0	1	0
DO≥4	0	0	2	1	3	1	1	0	0	1	2
DO≥5	0	0	2	1	3	6	2	0	1	1	2
DO≥6	2	1	3	1	3	8	5	0	0	1	2
DO≥5, BOD ≤10	1	1	4	1	3	3	2	1	1	1	2
DO≥5, BOD≤10, NH4-N ≤2	2	1	3	1	4	6	5	0	0	0	2
DO≥5, BOD≤10, Eff: NH4-N≤10	2	1	4	1	3	6	5	2	1	2	2
DO≥5, BOD≤10, Eff: NH4-N≤10, TP≤3	2	2	5	3	4	6	5	2	1	2	4
DO≥5, BOD ≤ 10, simul: with 50% ind.	1	1	4	1	3	3	2	1	1	1	2
DO≥5, BOD ≤ 10, simul: with 25% ind.	1	1	4	1	3	3	2	1	1	1	2
DO≥5, BOD ≤ 10, 50% ind.	1	0	2	1	3	2	2	0	0	1	2
DO≥5, BOD ≤ 10, 25% ind.	0	1	2	1	1	2	2	0	0	1	2

*Standards are expressed in mg/l

**First two letters of the name of the treatment plant (see Fig. 1)

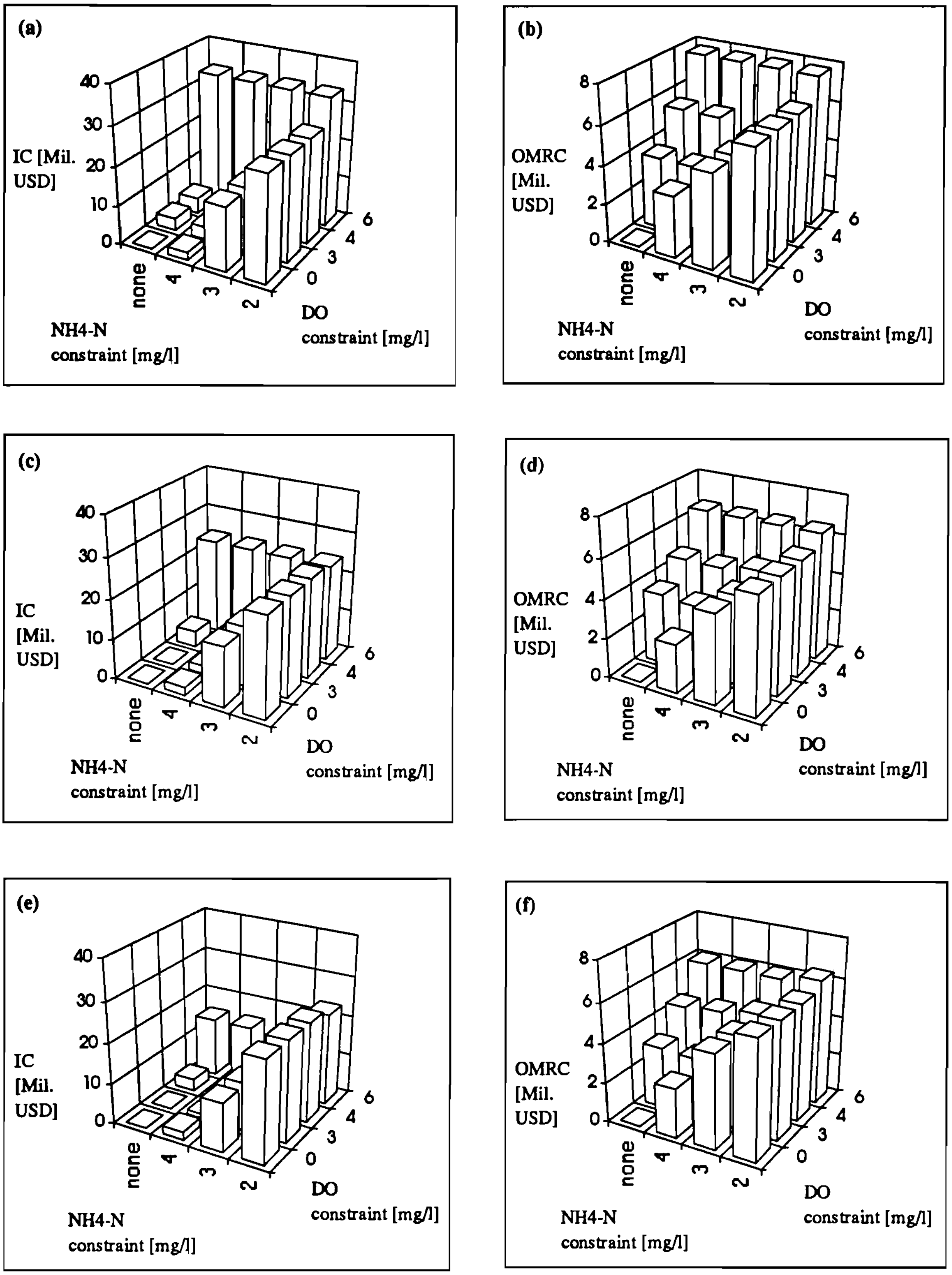

Fig. 7 Cost implications of setting DO and NH4-N standards under different industrial discharge scenarios. (a),(b) IC and OMRC respectively, corresponding to 1990 discharges; (c),(d) IC and OMRC respectively, for a 50% reduction of ind: discharges. (e),(f) IC and OMRC respectively, for a 75% reduction of ind: discharges.

• The role of industry is demonstrated by Fig. 7 and the last four lines of Table 2. Assumed emission reductions clearly improve DO and BOD conditions. However, the impact is reflected more pronouncedly by the costs. The "savings" (which in reality depends on the costs of industrial emission control) on the municipal emission control can be 13-14 million USD (see (10), (16) and (17)), expressing the need to develop an integrated least-cost strategy covering all the discharges of different natures.

• From the table, the selection of policy (7) or (8) seems to be logical, depending on the budget available (see Somlyódy and Paulsen, 1992 for a similar conclusion from a preliminary analysis of the Nitra problem).

4.5 The Impact of the Uncertainties of the Reaeration Coefficient on the Selection of Control Strategies

As indicated before, the reaeration coefficient (the most important model parameter) is subject to significant uncertainties (Masliev and Somlyódy, 1993). Thus, the question is, the extent that the derived policy can be influenced by these uncertainties. The issue is analyzed here in a deterministic, multiobjective, or regret framework (see for instance Burn and Lence, 1992, for a broader scenario analysis). The idea is to assume that in reality, the reaeration rate will differ from the design value (2 l/d in an average for our case). The difference then leads to a deviation in ambient DO quality. If the deviation is negative, standards will be "violated", but a positive deviation indicates overspending. The new treatment policies can be elaborated for the changed values of the reaeration rate. The cost-difference between the original design and the "real" one makes up either "missing" investment or "extra" spending, giving the idea of the monetary implications of the design "mistake". Large negative deviations of ambient DO criteria indicate high vulnerability of the treatment policy decision and, consequently, significant risks. If that risk is to be avoided, one can look for the "safe" solution, taking additional costs. Large investment margins, on the other hand, imply that "safe" policy decision amounts to high expenses. The regret analysis thus exposes decision risks and helps to determine whether the safe policy is affordable. Fig. 8 illustrates the minimum DO concentrations that may occur when the actual reaeration rate is different to the design values (the effect on nine designs done by assuming different K_r values and setting various DO constraints are presented). Each curve of the Fig.8 correspond to one set of design conditions. The point corresponding to the design conditions are indicated on each of them. The other four points on each curve were obtained by subsequent simulations. For these simulations, the set of "optimal" treatment alternatives found for the respective design condition was assumed to be in operation. The simulations was performed for the four different K_r values, in order to obtain the four other points of the curve. It is observed that the curves of two of the design conditions are coinciding (designs conditions: K_r=2, DO_{min}=5.6; and K_r=1.5, DO_{min}=5). That implies the possibility to choose the cheaper one (in this case K_r=2, DO_{min}=5.6), when considering a selection from those two alternatives.

A summary given in Table 3 outlines the regret analysis for the treatment policies developed for three target DO ambient criteria. It is assumed that the actual reaeration rate K_r can differ from the mean by +/- 0.5 l/d (which corresponds to about +/- 2 standard deviations, see above). The total range of K_r thus defined is broader than suggested by the uncertainty analysis performed earlier (see Masliev and Somlyódy, 1993). The first number in each cell of the table is the difference in IC in million USD. "Missing" investments are indicated by negative signs, and positive figures show "extra" expenses. The second figure is the change of DO criteria in mg/l, negative signs showing the violation of the target ambient standard.

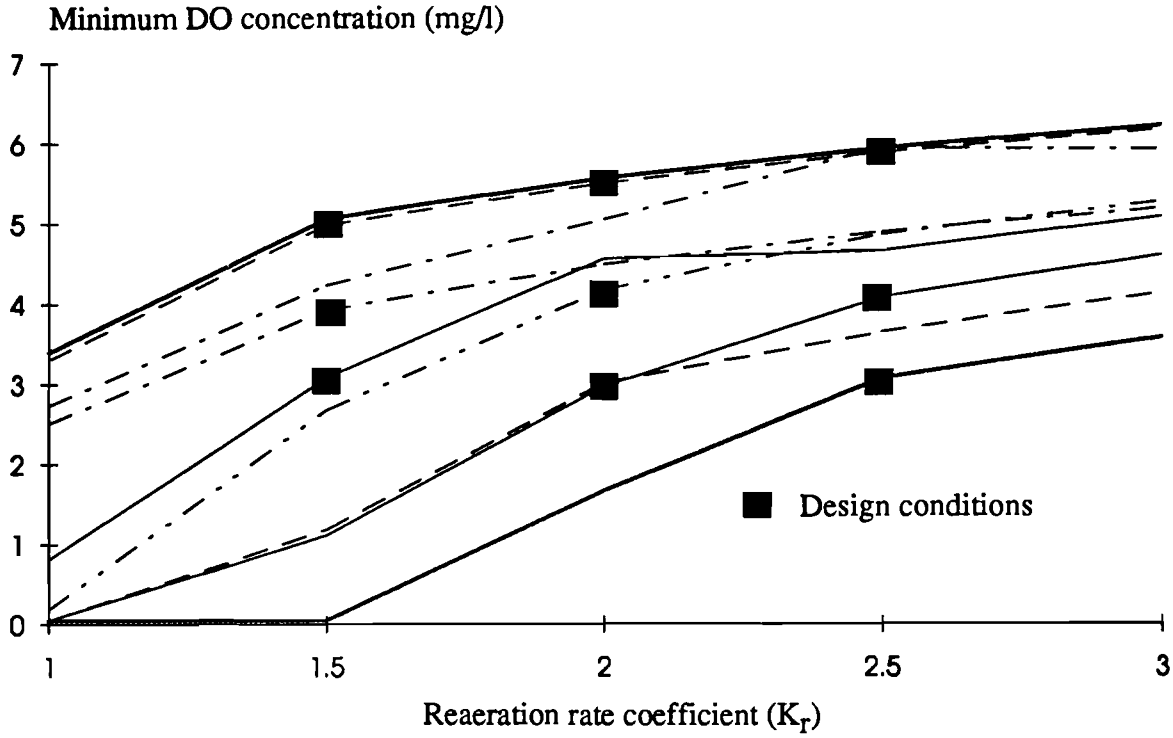

Fig. 8 Minimum DO concentrations that may occur when the actual K_r value is different to that considered in the design

TABLE 3 The Role of Uncertainty of the Reaeration Coefficient: Its implications on the Investment Cost and the Ambient Oxygen Concentration

Actual K_r[1/d]	DO≥3 Design K_r[1/d]			DO≥4 Design K_r[1/d]			DO≥5 Design K_r[1/d]		
	1.5	2.0	2.5	1.5	2.0	2.5	1.5	2.0	2.5
1.5	0/0	-1.0/-1.8	-4.2/-3.0	0/0	-17.0/-1.5	-19.5/-3.0	0/0	-16.4/-0.8	-24.2/-2.1
2.0	1.0/1.5	0/0	-3.2/-1.4	17.0/0.6	0/0	-2.5/-1.1	16.4/0.5	0/0	-7.8/-0.6
2.5	4.2/1.6	3.2/0.6	0/0	19.5/1.0	2.5/0.7	0/0	24.2/0.9	7.8/0.5	0/0
IC	4.2	3.2	0.0	21.0	4.0	1.5	31.4	15.0	7.2

The table also illustrates the high sensitivity of investment cost to the value of K_r (the bottom line). A higher K_r leads to increased natural O_2 input to the river water, and thus, a lower level of treatment is necessary.

As can be seen from the table that the drop in the DO level caused by the reduction in the reaeration rate exceeds the raise under equal parameter increase. This occurs due to the saturation character of the problem (the upper limit of DO_{min} under municipal emission control is less than 6 mg/l, see Table 2.) Similarly, missing investments exceed overexpenditures at higher DO levels.

For the DO criterion of 3 mg/l, the vulnerability of ambient water quality is high, while possible overexpenditures and additional investment needs are small. This calls for a safe policy decision to invest 4.2 million USD which guaranties DO above 3 mg/l.

For target DO levels 4 and 5 mg/l, the over and underexpenditures become much higher. The smaller deviations of the DO criteria makes the policy selection even more difficult. Interestingly, the policies aimed at DO level 4 mg/l do not look too attractive and the DO=5 mg/l strategy of 15 million USD investment is perhaps the best compromise. The additional investment requirement may be rather high if K_r=1.5 l/d is "realized", but the respective improvement in DO level (from 4.2 mg/l to 5 mg/l) is not in proportion with the expenses. Thus, the above policy can be considered as relatively cheap one which (as contrasted to the present DO_{min} level around 2 mg/l) guarantees DO between 4.2 mg/l and 5.5 mg/l (and the "worst" DO levels may be observed under low flow conditions only, occurring for at most one or two weeks in a year).

Finally, DO=6 mg/l strategies (not shown in Table 3) do not offer too many interesting features. They are expensive (25-35 million USD, depending on K_r) and safe (DO=5.5-6.4 mg/l, K_r=2 l/d). In addition, over or underexpenditures are smaller (10 million USD) than for the DO=5 mg/l case.

The scheduling of control policies under severe limitation of financial resources can be done by first implementing the DO=3 mg/l, IC=4.2 million USD strategy. It can then be upgraded through the 5 mg/l (IC=15 million USD) plan to a DO=6 mg/l policy later on (as the economic situation improves). The idea of multi-stage development of WWTPs can be applied for ensuing the smooth and inexpensive upgrade path (see Somlyódy, 1993, for details).

5 CONCLUSIONS

(1) Dynamic programming is well suited to handle river basin water quality management problems generically. It allows to incorporate simulation models (linear or non-linear) expressing the impact of emissions on ambient water quality, and to consider the details of alternative waste water treatment technologies. Possible extensions to the optimization process include the incorporation of parameter uncertainty by a variance-based approach or by a scenario analysis. Different formulations of the problem, in terms of objective functions and constraints, can also be used to assess the robustness of the strategies formulated. Incorporation of quality indicators other than BOD, DO, N and P is necessary to model the various processes that are occuring in a river more accurately. Scheduling of the control activities over the planning horizon is another important problem to be solved.

(2) For the Nitra River, the range of realistic expenditures was extremely broad (between 3 and 95 million USD) depending on the policy formulation (based on effluent standards, ambient standards or their combinations) which indicates significant saving possibilities. The most

expensive solution is to replace all the treatment plants with new ones satisfying the most stringent recommendations of the European Community. The present (and future) Slovakian effluent-quality standard system implies an investment of 32-35 million USD. A least-cost policy leading to Class III water (in terms of DO, BOD and NH4-N) is roughly equivalent with the former one. The water quality is identical for all three cases.

(3) Since control actions influence several constituents simultaneously, least-cost policies developed solely on the basis of DO ambient criterion lead to significant improvement with regard to other components as well. These strategies are attractive: an investment of 4-15 million USD improves DO from about 2 mg/l to 3-5 mg/l. A multiobjective analysis showed that these policies are not too vulnerable to uncertainties in parameters of the DO model used within the optimization. The possible overexpenditures are not high either. As contrasted to effluent standard based strategies, the least-cost ones are rather non-uniform. A long-term policy can be obtained by a sequence of least-cost strategies under gradually tightened ambient quality standards.

(4) The economic consequences of setting standards are strong and non-linear. The improvement of DO_{min} from 4 mg/l to 5 mg/l requires more than 10 million USD investment, while BOD and NH4-N also play a similar role.

(5) In the present study, only municipal emissions were directly incorporated into the optimization (which represent 70% of the total BOD discharge into the river system). A sensitivity analysis demonstrated the significant role of industrial pollutant loads, which clearly calls for the development of an integrated least-cost policy covering all the emissions of various origins.

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